Some factors governing the water quality of microtidal estuaries in South Africa

BR Allanson
Institute for Water Research, Rhodes University, Grahamstown 6140, South Africa

Abstract

The role of coastal geomorphology and Man-made alterations, including reduced river flow through dam construction, determines, at least in part, the water quality of South African microtidal estuaries. To offer increased understanding of the manner in which these features may modify water quality, a short description of the biogeochemical processes in estuaries is provided. Comment on the present limitations of modelling some of the estuarine processes in South African investigations is given.

Nomenclature

aeolian	transported by wind
DO	dissolved oxygen
DOC	dissolved organic carbon
DON	dissolved organic nitrogen
DIN	dissolved inorganic nitrogen
DOP	dissolved organic phosphorus
DIP	dissolved inorganic phosphorus
EFR	estuarine flow requirement - pertaining to required river flow
Eh	redox potential expressed in millivolts
MAR	mean annual rainfall
MSL	mean sea level
N:P	normally the value of the ratio of total soluble nitrogen to total soluble phosphorus in the water column
ppt	parts per thousand - with respect to seawater, meaning kg of salts per kg of solution.
POC	particulate organic carbon
POM	particulate organic matter
PON	particulate organic nitrogen
SRP	soluble reactive phosphate
TDL	theoretical dilution line which represents a linear dilution of a solute as it passes through the estuary - such solutes are termed 'conservative'.

Introduction

The earliest synthesis which described the water quality of South African estuaries was prepared by Day (1981). It showed that while the principles of estuarine chemistry established in the Northern Hemisphere were applicable to southern African estuaries, there were sufficient differences in the hydrodynamic features of these estuaries which could influence water quality and the like. It is the purpose of this paper to examine those features of coastal geomorphology which define (at least in part) the hydrodynamic properties of the estuaries, and how they may affect the water quality of a number of representative estuaries along the South African coast.

☎ (044) 384-0658; fax: (044) 384-0658; e-mail: ba11@pixie.co.za
Received 12 June 2000; accepted in revised form 23 March 2001.

This approach has been adopted in order to highlight those geomorphological features which should be incorporated into the management strategies for estuaries in South Africa. In so doing, we become increasingly aware of the ecological and socio-economic linkages which are undergoing both spatial and temporal co-evolution (Crooks and Turner, 1999). It is also necessary to recognise that upon this morphological template changes due to varying coastal processes are imposed which may further alter both structure and hydrodynamics and, therefore, water quality.

The influence of Man-made alterations on estuarine structure and hydrodynamics, brought about by bridges and dams, and the likely impact of the transfer of impounded water between catchments are described. Further, and in view of the overwhelming importance of inter-tidal wetlands in the carbon, nitrogen and phosphorus metabolism of estuaries, the general principles of these pathways are described, and their role in South African estuaries, particularly with respect to handling soluble wastes of human origin, are commented upon.

Some geomorphological features

Recent studies in South Africa have demonstrated the modulating influences brought about by geomorphological processes in shaping the coastal scenery since the Pleistocene (Cooper et al., 1999). Of particular significance was the substantial fall in sea level at the height of the Flandrian glaciation (18 000 BP) which exposed the Agulhas Bank to the edge of the continental shelf. This, coupled with the strengthening of onshore winds across the exposed coastal zone due to the increasing differential between sea and land temperature (Hobday, 1979), brought about substantial aeolian transport which built up extensive dune cordons. In KwaZulu-Natal the dune cordons from Mtunzini northward into Mozambique constitute one of the largest persistent cordons in the world (Hobday, 1979). Dunes with crests elevated to 200 m are rarely interrupted by estuaries, but lakes formed by sealing of the palaeo-estuaries are common, and the largest of these is Lake Sibaya, South Africa (27° 22' S : 32° 41' E).

Along the southern coastal rim-land, the geomorphology is dominated by the ancient Cape Fold Mountains, which lie parallel to the coast, resulting in a series of inward-moving rain-shadow areas of increasing severity. The rim-land has been subject both to uplift and to the later effects of variation in Pleistocene sea level. The consequences of sea level regression and subsequent transgression are clear to see in a series of coastal embayments, for example, the Wilderness Embayment. The present structure of which has been further defined by high dune cordons built during these Pleistocene events. In adjacent parts of the coast, deeply incised valleys formed by down-cutting by the coastal rivers of the Tertiary coastal platform end in narrow steep-sided estuaries in which flood tide deltas are variously developed. In KwaZulu-Natal uplift of the Natal Monocline (King, 1972) and the consequent rejuvenation of major rivers eastward to the sea has caused the rivers to gouge out mighty valleys. The Tugela River valley is 1 000m deep at Kranskop, 50 km from the sea.

This entrenchment of rivers has, during the Pleistocene marine transgression, resulted in drowned valleys. And with the continual erosion of the hinterland, their estuaries have become filled with terrigenous debris. Further modification occurred during the Holocene when the impact of climate upon estuarine environments was of paramount importance. The aeolian transport of sands resulted in segmentation and infilling of coastal lakes and estuaries. Orme (1973) has recorded that the area of the lakes in KwaZulu-Natal has been reduced by some 60% during the Holocene.

During the past 100 years human activities in South Africa have accelerated the natural progression of change in estuarine geomorphology, either by reducing river inflows by dams or by excessive water abstraction, which in extreme cases (Whitfield and Bruton, 1989) can lead to the total destruction of the estuary, or to consolidation of estuarine sediments (Reddering, 1988) and an increasing frequency of bar formation and extended closure of the estuary.

Influence of geomorphological structure

It is appreciated that by adopting this approach, criticism of the rigidity it imposes is likely (Pethick, 1984). However, structure is a convenient framework within which to explain other more pertinent geo-biological events, while recognising that, within the array of microtidal estuaries in South Africa, fairly clear geomorphological types exist.

Estuaries formed within incised valleys with limited or no flood plains

A particularly well-worked example is the Palmiet Estuary (34° 21' S :19° 00' E) in the Western Cape Province. The river catchment is principally made up of Table Mountain Sandstones covered by a mosaic of orchards and fynbos, and because of the underlying sandstone the waters are poor in nutrients and calcium but rich in humic substances derived from the natural plant cover.

The estuary is subject to large inflows due to winter rains, and Largier and Slinger (1991) and Largier and Taljaard (1991) have described the hydrodynamic structure of this incised prototypical bar-built estuary under high flow and low flow conditions. The high river flows during winter create a marked fjord-like salinity stratification in the estuary. The hydrodynamic description requires that the intrusion of sea water be controlled by internal hydraulics at the mouth. A saline wedge is spread into the estuary as a non-mixing density current. Vertical transport is controlled by stratified shear flow and salinity removal in the fast-flowing surface layer: A consequence is the persistence of a long-resident deep saline layer during summer and to a lesser extent in winter. The overall circulation which Largier and Taljaard (1991) describe is similar to that observed in Alaskan fjords (e.g. Glacial Bay), except that in the Palmiet Estuary the time scale is based upon the spring-neap tide cycle, while in Glacial Bay the bottom water is renewed annually.

It is to be expected that the chemical quality of the water column will reflect this dynamic state. Allanson and Winter (1999), in a review of the general chemical properties of South African estuaries, records from the data of Taljaard and Largier (1989) that, during a winter spring tide cycle with river flows varying from 9 -15 $m^3{\cdot}s^1$ and flood tide volumes from 20 to > 60 m^3, the differences in pH, SRP, nitrate-N, nitrite-N, DON and DOC between surface and bottom indicated that a pronounced chemical stratification existed. During a neap tide cycle when the river was flushed with >30 $m^3{\cdot}s^1$ the tidal range was depressed, and there was no evidence of flood-tide volume and all parameters were similar throughout the water column. Thus, the river is the nutrient source of the surface layer during winter, while the sea provides a source for the bottom layer during summer.

Both Taljaard (1987) in the Palmiet Estuary and Allanson and Read (1995) in the Keiskamma Estuary report on the formation of insoluble complexes within the DOC/DON pool at the head of the estuaries under increased ionic activity. The role that these flocculates play in the chemical cycles of the estuaries is uncertain, and if we add to this the net imported POC of river and marine origin, estimated by Branch and Day (1984) for the Palmiet Estuary to be 143 000 $kg{\cdot}yr^1$, the allochthonous sources of reduced nitrogen and carbon are substantial.

In an estuary with a restricted intertidal area, these allochthonous subsidies represent an essential energy source for the resident micro- and macro-benthic fauna. But notwithstanding these inputs, the quantity of organic matter in the sediments of the Palmiet Estuary remains low due to the high winter flows, and this increases the coarseness of the sediment, which, in turn, prevents the accumulation of organic material. Some biological activity is due to the burrowing sand prawn, *Callianassa krausii*, which in the more stable lagoon-like section concentrates fine particles and organic material in the lining of its tubes. But, overall, the impression gained is that, in summer, the ephemeral stock of the alga *Cladophora* spp, acts as a sink for nutrients upon which the deposit feeders are dependent.

Management of this estuarine type will demand careful consideration of these principles, and in particular, that its ecological operation is dependent to a large degree upon external energy sources (carbon) and nutrients (N and P), both riverine and marine. Impoundment of the main stem river will lead to a severe reduction in freshwater and allow the sea to dominate the hydrodynamics and chemistry. Alternatively, enrichment of the river inflow by anthropogenic activity is likely to reduce nitrogen limitation on phytoplankton or macroalgal growth, leading to persistent and harmful bloom formations.

Estuaries with relatively extensive flood plains

Within the spectrum of South African estuarine geomorphological structure, such systems are common. They usually, but not invariably, exhibit gently sloping intertidal shorelines on which a wetland plant association is developed. These areas are essential in the modulation and transfer of inorganic and organic inputs (vide p 18) and are richly dissected by tidal creeks which serve to dissipate the energy of the tidal flow (Pethick, 1992).

River flow or the lack of it contributes further to the water quality of all estuaries, and particularly of this type, which may range from those with marked seasonal axial gradients in salinity, as in the Great Berg Estuary, to those in which the dominant forcing mechanism is the diurnal tide, for example in the Kariega Estuary. A new variant has been created through the influence of inter-basin transfer of freshwater. An example is the Great Fish Estuary

(Grange and Allanson, 1995; Grange et al., 2000). This estuary has changed from one which experienced long periods of drought to one with a sustained turbid inflow as a result of the transfer of water from the Orange River into the upper Fish River catchment. All tend to show partial stratification of the water column.

Our limited data set indicates that mixing of the major dissolved constituents (Na, K, Ca, Mg, Cl and SO_4) is conservative, and because of the 'neutral' position of these cations and anions in the chemistry of the water column, by far the greatest research effort has been put into describing the varying nutrient status of these estuaries. However, nutrients such as nitrate may also act conservatively depending upon the flow-through time of the estuary.

The Great Berg Estuary (32° 46' S:18° 09' E) is 45 km long with strong seasonal flows. During winter water quality is influenced by river flow with its high levels of nitrate-N (504 $\mu g \cdot \ell^{-1}$) due mainly to agricultural activity in its catchment and near oxygen saturation. A flow maximum of 389 $m^3 \cdot s^{-1}$ in September has been reported (Slinger andTaljaard, 1994). Under these conditions, tidal intrusion is limited to about 10 km from the mouth, while during summer, river flow even as low as 0.5 $m^3 \cdot s^{-1}$ is critical in limiting the upstream migration of sea water, emphasising the importance of sustaining the base flow of the river.

Slinger et al. (1998) have reported that, apart from SRP, nutrient concentrations were linearly related to salinity during the high winter flows. For example, on 25 August 1995 dissolved nitrite - N and nitrate - N gave R^2 values of 0.96 and 0.88 respectively. The authors concluded further, that the major determinant of water column nutrient dynamics under high flow conditions is the tidally-driven circulation and mixing rather than processes of regeneration or depletion. Decrease in winter flows allowed the re-establishment of partial stratification in the middle section of the estuary, and the increase in SRP at this time was due to regeneration, which Slinger et al. (1998) interpret as relating to the limited renewal of sea water in the estuary via flood-tide intrusion. An N:P ratio of 1.7 :1 has been reported by Slinger and Taljaard (1994) for the summer regime so that the estuary in common with other South African estuaries, e.g. Knysna and the Kariega, is considered to be nitrogen-limited, (Allanson and Winter, 1999).

Prior to engineering restructuring in 1996, Slinger and Taljaard (1994) point out that the estuary opened to the south and was shallow during low flow conditions, owing to the inward transport of marine sands. This shoaling of the entrance would have constrained the extent of the axial tidal influence, so that the present hydrodynamic features may well be the result of this Man-made perturbation.

Contrasted with this major west coast system is the Swartkops Estuary (33° 57'S :25 °28') near Port Elizabeth on the south east coast. The estuary has an extensive flood plain and is partially stratified, and has a relatively large tidal prism (3.06 x 10^6 m^3). Throughout its tidal length, the littoral of the estuary and its near catchments have been subject to major urban development. The detailed investigation of the impact of urban run-off on the water quality of the estuary by MacKay (1993) has stressed the importance of understanding the impact of Man-made structures upon tidal transport within an estuary if any sensible and definitive statements are to be made about the fate and impact of polluting inputs.

The estuary is divided into an upper estuary by the levees of the Wylde Bridge 3 km upstream, and a seaward marine embayment. The embayment has strong currents and good flushing, while the estuary above the bridge has the hydrodynamic characteristics of a bar-built estuary. This effective translocation of an estuary mouth upstream as a result of a man-made structure has important consequences with regard to the retention of pollutants in the area above the second mouth, particularly at neap tide. MacKay (1993) has established that during spring tides the upper estuary will experience some exchange with new seawater. At neap tides, the lower estuary is not entirely flushed so that new seawater is unlikely to gain access to the upper estuary. Superimposed upon this altered tidal flow regime are the periodic discharges of three polluting sources which occur above the bridge. These were found to affect the water quality of the estuary for 10 to 14 d, and after flooding up to one month was required for salinity to return to pre-flood levels.

Subsequent investigations, by Slinger et al. (1998), of the relationship between salinity and the dissolved nutrients, e.g. DIN and SRP for river inflow varying from 0.4 $m^3 \cdot s^{-1}$ to 6.4 $m^3 \cdot s^{-1}$ during spring tide, showed that the nitrogen components did not exhibit conservative behaviour while SRP and, to a lesser extent, silica did. The total inorganic nitrogen, although elevated in the river at the tidal limit (200 to 1500 $\mu g \ell^{-1}$) augmented along the tidal reach as concentration levels appear to be sustained, notwithstanding increasing estuarine volume. This is comparable with the behaviour of these compounds in the Great Berg Estuary, and suggests that, as regards the nitrogen components, whether they derive from agricultural or human waste disposal, their behaviour is similar.

The conservative behaviour of SRP in both estuaries is likely linked to an intrinsic buffering mechanism typical of estuaries in general (Froëlich, 1988). However, in the Swartkops Estuary the high levels of SRP (84 to 179 $\mu g \ell^{-1}$) are so greatly in excess that any combination with iron, adsorption onto particulates or involvement in biochemical processes within the intertidal mudflats are masked. A somewhat different interpretation is provided by Scharler et al. (1998), who suggest that the decrease in SRP towards the mouth is seen as a net uptake of phosphate. Nevertheless, since the concentrations are high, export to the sea is substantial; 24 $t \cdot a^{-1}$ are estimated to be delivered to Algoa Bay.

The effect of drought on the supply of nitrogen, phosphorus and silica in estuaries is illustrated by the the mixing plots (Allanson and Winter, 1999) of the Kariega and Great Fish Estuaries, east of the Swartkops Estuary (Fig. 1a,e). When river flow is sustained, river-derived nitrate-N behaves conservatively, but during drought when river flow stops, as in the Kariega River, the sea becomes a dominant source of new nitrogen (Taylor,1992). The mixing plots of SRP in both the Kariega and Great Fish Estuaries (Fig.1b,f) depart from the theoretical dilution line provided that river flow is sustained. This implies a phosphate source in the middle of the Kariega Estuary and a sink in the upper Great Fish Estuary. Under conditions of severe drought in the Kariega Estuary, as in December 1984, (Fig. 1c), a reversed salinity gradient was established and SRP increased linearly with salinity. Silica concentrations in both the Kariga and Great Fish Estuaries are elevated above the TDL indicating a source in the estuaries.

Estuaries arising from barrier lakes

A somewhat different scenario is created in those estuaries in which river flow over a coastal plain is interrupted by large or small barrier lakes which overflow into the estuaries

There is an array of estuaries which fit this geomorphological classification. The Swartvlei Estuary, at Sedgefield, is a particularly good example, while another is that of the Kosi lakes system (26° 54' S : 32° 48' E) near the north-eastern boundary with Mozambique. An extreme case where salinities fluctuate widely as a result of floods and droughts is shown by the extensive St Lucia wetland system in KwaZulu-Natal. Smaller systems are, for example, the Klein River lagoon at Hermanus (34° 25' S :19°18' E), (Scott et al.,

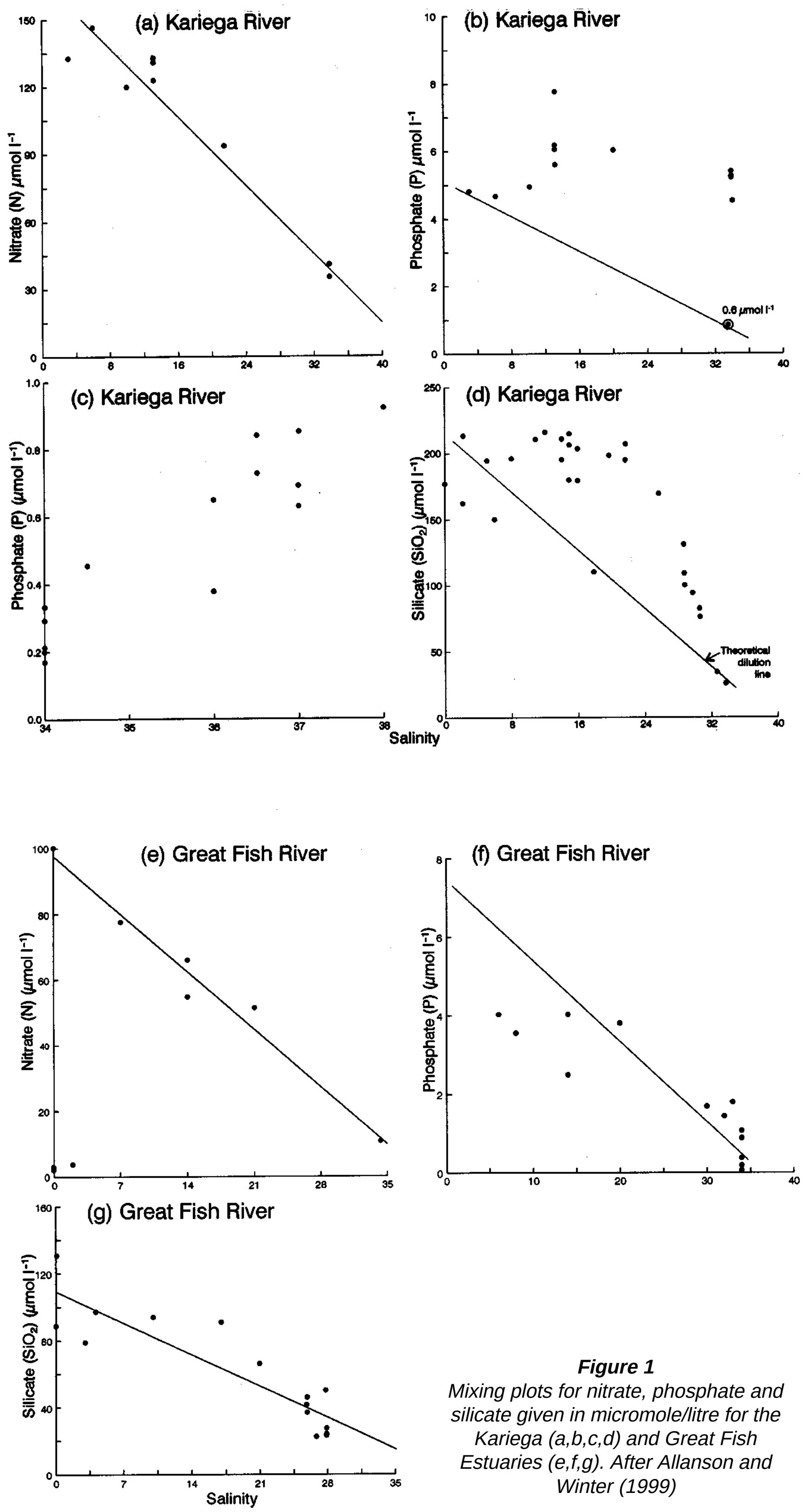

Figure 1

Mixing plots for nitrate, phosphate and silicate given in micromole/litre for the Kariega (a,b,c,d) and Great Fish Estuaries (e,f,g). After Allanson and Winter (1999)

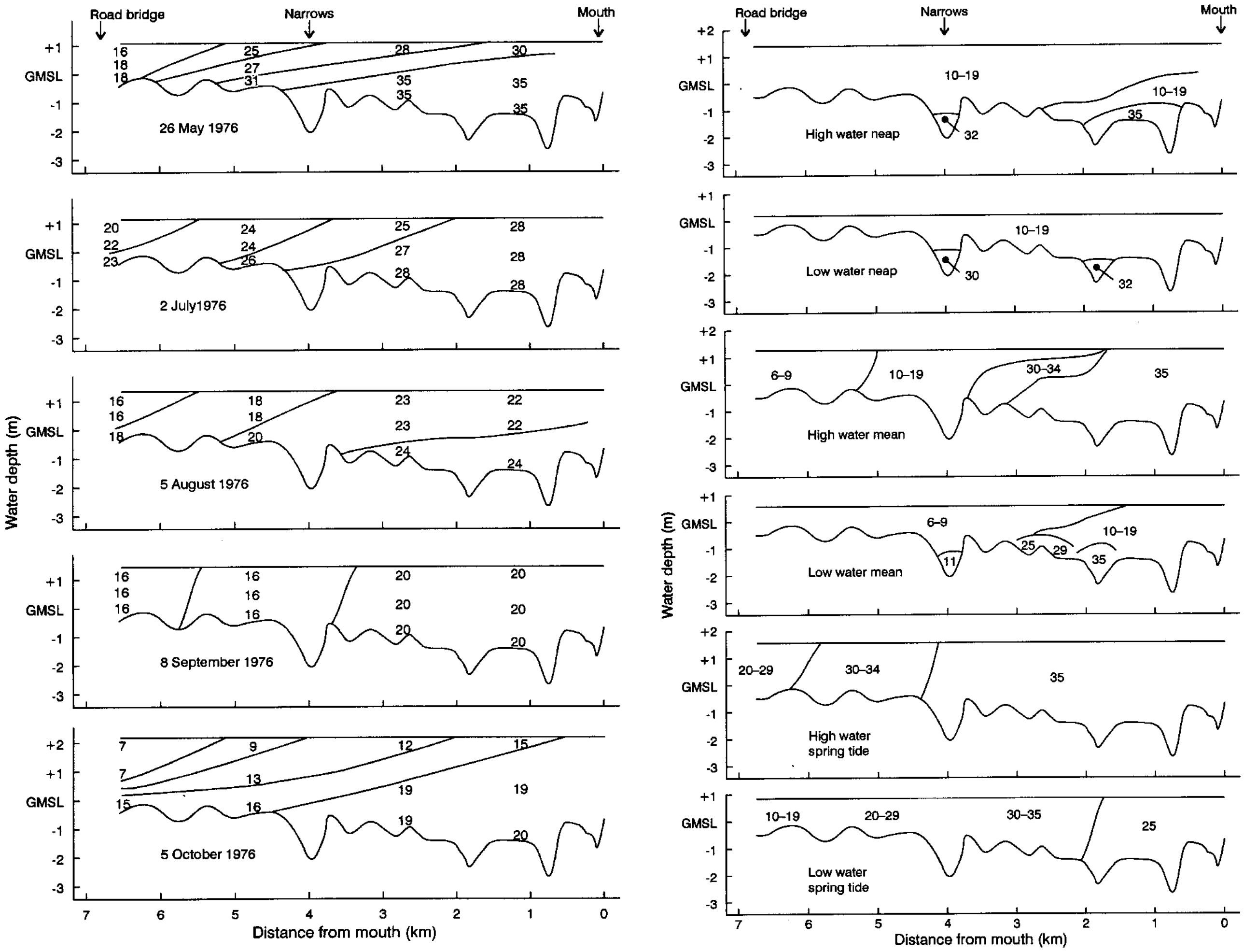

Figure 2

Salinity profiles during (a) lagoon and (b) at neap, mean and spring tides in the Swartvlei Estuary . After Liptrot (1978).

1952) in which freshwater inflow is balanced by evaporation so that normal salinity gradients are maintained. In contrast, in the Diep Estuary (32° 53' S :18° 28' E) evaporation in the upper lagoon exceeds inflow so that hypersalinity develops (Taljaard et al., 1992).

Against this background it is difficult to decide what might be considered reasonably representative, but the Swartvlei Estuary and the barrier lake, Swartvlei at its head were chosen because they show a number of features which are characteristic of this geomorphological type, notwithstanding the impact of rail and road bridges and a small resident urban community.

The Swartvlei Estuary (34° 00' S : 22° 46' E) is a mature shallow sinuous system which is frequently closed during the winter months, and opened almost always by human intervention in the early spring to prevent flooding of low-lying properties and their septic tanks! Salinity profiles for the estuary during lagoon and tidal conditions are reported in Fig. 2a,b. The dilution of the estuary from May to October by brackish water from the slowly rising Swartvlei at the head of the estuary results in weakening of the isohalines and a 50% reduction in the overall salinity gradient of 10 to 19 ppt (Fig. 2a). It is particularly significant that this gradient is maintained in the lagoon phase as it reduces environmental stress on plant and animal communities which, under either freshwater or marine conditions, would disappear. With the onset of spring rains, the mouth is breached and tidal penetration, which increases from neap to spring tide, generates a new salinity gradient of 6 to 34 ppt, (Fig. 2b). Pockets of dense water held in the scour holes during winter are removed by entrainment in the tidal flow.

During the lagoon phase, DO falls to zero in the bottom water and organic carbon rises to >40g·m^3 as a result of the decomposition of *Zostera* and *Enteromorpha* in the scour pools. Nitrate - N concentrations decrease (Allanson and Winter, 1999) and almost invariably the scour pools exhibit low Eh, and bubbles of hydrogen sulphide and precipitated sulphur are frequently observed. The evolution of hydrogen sulphide taints the atmosphere above the lagoon, giving rise to the well-known Sedgefield "stink".

Estuarine lakes

Estuarine lakes have retained something of the depth to which river channels eroded during the Pleistocene. Their floors are often well below sea level and covered with gytta-like sediments. The basins are filled with sea water, which is only rarely exchanged, and separated from a surface brackish layer by a steep halocline. Such systems are termed *meromictic*. The Msikaba Estuary (31° 18' S : 29° 58' E) is a good example with a recorded depth of 35 m. Wooldridge (1976) has provided a useful description of salinity and temperature profiles. The halocline is at 1 to 2 m; thereafter the salinity is uniform to the analysed depth of 23 m.

Robarts and Allanson (1977) have described the seasonal or

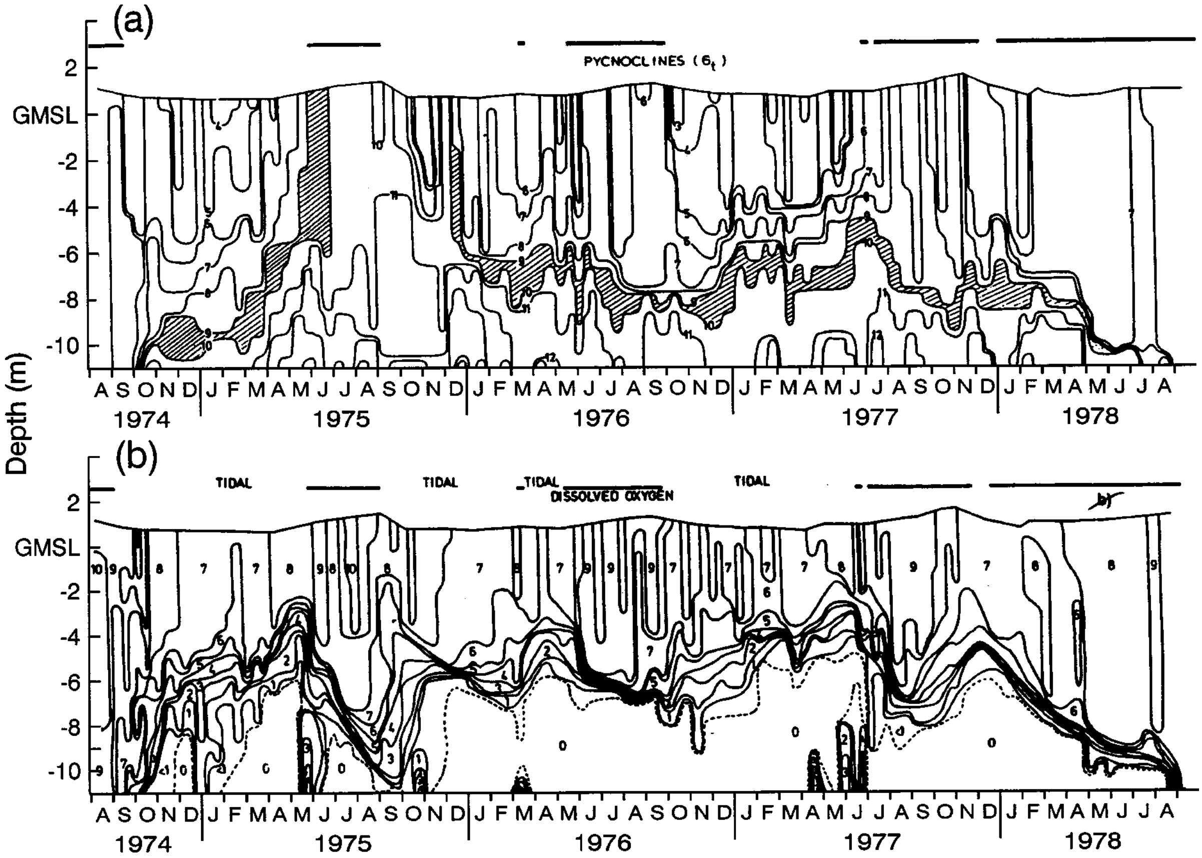

Figure 3

Pycnocline (density) (a) and dissolved oxygen (b) isopleths in Swartvlei between 1974 and 1978. The horizontal bars represent periods when the mouth of the estuary was closed. After Allanson & Howard-Williams (1984).

cyclical meromixis in Swartvlei while Allanson and Howard-Williams (1984) have described its hydrodynamic features in more detail. The meromixis is linked to the status of the mouth of the estuary. During periods of tidal penetration into the lake a pronounced halocline is established at between 5 to 6 m (Fig. 3a) with consequent loss of oxygen and an increase in hydrogen sulphide levels in the deep water as long as the tidal flow reinforces the density difference, (Fig. 3b). Closure of the mouth, particularly in autumn and winter, which are the windiest times of the year, results in the gradual erosion of the halocline, (Fig. 3a,b), by wind-induced turbulence until oxygen reaches the bottom, -11m MSL. The cycle is repeated once tidal conditions are re-established.

An example of permanent meromixis is provided by Lake Mpungwini in the Kosi Estuary system. This meromixis was first described by Allanson and Van Wyk (1969) who showed that the brackish water flow from the upper lakes remained at the surface, being prevented from mixing into the lake by the deep density layering introduced by tidal flow. Later Ramm (1992) established that the high levels of hydrogen sulphide below the halocline were due to the dissimilatory reduction of sulphate mediated by bacteria:

$$\text{organic carbon} + SO_4^{2+} \rightarrow CO_2 + \text{sulphides}$$

and confirmed that the increased concentrations of chloride and sulphate below the halocline reflect the intrusion of more dense sea water. When molar ratios of sulphate-sulphur to chloride were computed, sulphate decreased with respect to chloride below the halocline, indicating the reduction of sulphate. If sulphide-S is added to sulphate -S, the ratio of both forms of sulphur and chloride remains relatively constant with depth.

As the surface waters of this lake provide the only access to or from the sea to the upper lakes, contamination by free sulphide through upwelling of bottom water would become an effective barrier, preventing upstream or downstream migrations of fish and causing mortality of resident species.

An opportunity to test the predictive ability of this model arose in September 1989 during equinoctial spring tides. These tides were followed immediately by a rapid shift in wind direction. High winds from the north-east ($4\ m \cdot s^{-1}$) backed to the south-east when wind speeds increased to $6\ m \cdot s^{-1}$. On both occasions high concentrations of sulphides (>10 $mg \cdot \ell^{-1}$) upwelled to the surface, and the largest fish kill ever reported in a KwaZulu-Natal estuary took place.

These meromictic lakes are delicately poised, and wastes of anthropogenic origin which are allowed to enter such systems, increase the likelihood of severe pollution when tidal flow and freshwater inputs are minimal.

Influence of man-made development

We now examine the impact of such development upon one or other geomorphological type. Those which have been most seriously affected so far have major dams constructed on the main stem river, comparatively near to ebb and flow and the estuarine flood plain.

Major and minor dams

These represent the most critical of human impacts upon the geomorphology (see p. 4) and water quality of estuaries. South Africa has one of the lowest mean annual precipitation: mean annual runoff (MAP: MAR) conversion ratios in the world, 8.6% (Alexander, 1985) and with only two small natural lakes, river water must be conserved at all costs. Simplistic attempts have provided first-order estimates (Jezewski and Roberts, 1986) of the evaporation requirements and minimum flood requirements of estuaries and lakes. They have, however, been found to be some 8% of the total exploitable water resources of the Republic and this demand may become a considerable constraint upon the available resource. Fortunately, the National Water Act 36 of 1998 has recognised the intrinsic worth of estuaries, and now requires that a reserve be determined for estuaries: the reserve being defined as the quantity and quality of water required to satisfy basic human needs and to protect aquatic ecosystems in order to secure ecologically sustainable development and use of water resources. Prior to the development of the reserve principle, an EFR, which specifies the quantity of water required, had to be determined where developments within the river catchment or on the floodplains were likely to alter river inflow to the estuary. It is, therefore, likely that with the determination of the reserve (or an EFR), the overall water resource can be wisely used in the maintenance of estuarine water quality.

The application of the reserve principle and the EFR is still at an early stage. Nevertheless, there have been a number of useful attempts to understand how estuaries respond to reduction in freshwater inflow, and how the impact may be ameliorated by proper management of flow from the upstream dam(s).

A recent whole estuary experimental study by Slinger et al. (1995) on the Great Brak Estuary (34° 02' S : 22° 14' E) has begun to unravel the effect of varying freshwater flow upon hydrodynamic behaviour and water quality of partially mixed estuaries. Prior to a pre-determined freshwater release from an upstream dam, and breaching of the mouth (Fig. 4a), salinity exceeded 20 ppt, and the water column was stratified. Temperature stratification was also evident and lowered DO occurred in the deeper layers (1.5 $mg \cdot \ell^{-1}$) while anoxic conditions were measured at the bottom of the deepest sampling station (-3m MSL). This was associated with elevated ammonium-N (568 $\mu g \cdot \ell^{-1}$), SRP (66 $\mu g \cdot \ell^{-1}$) and reactive silica (1 146 $\mu g \cdot \ell^{-1}$). In other respects, the levels of dissolved nutrients were within the range of marine and freshwater sources (Slinger et al., 1995). Following the release of freshwater equivalent in volume to the tidal prism and immediately prior to breaching of the mouth, a wedge of low salinity water (<5 ppt) formed along the surface of the estuary (Fig. 4b). Once the mouth was breached this lowered salinity surface water was selectively withdrawn while the estuary drained down to +0.79 m MSL. The older and deeper saline water remained trapped

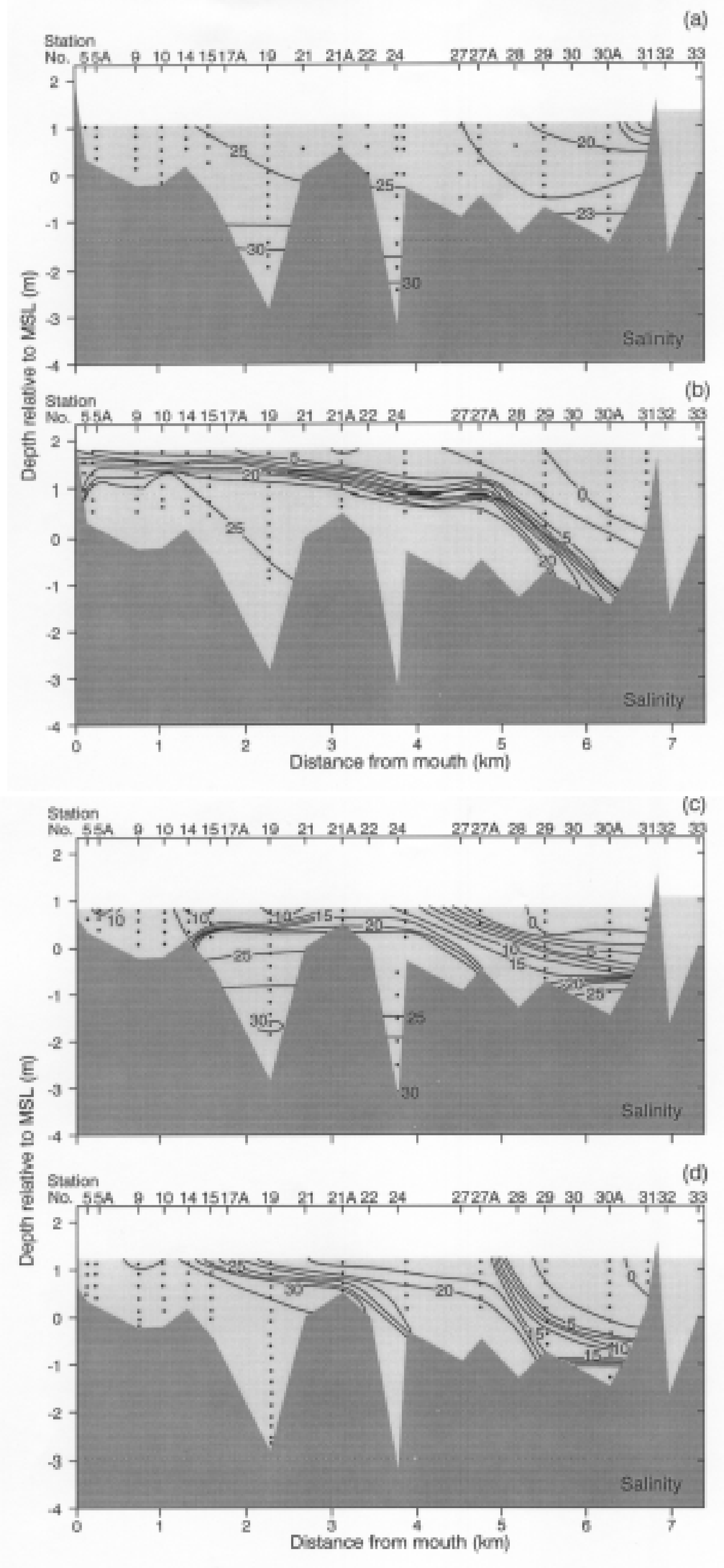

Figure 4
Salinity profiles in the Great Brak Estuary (a) prior to flooding; (b) after the planned flood but before the bar was breached; (c) during late ebb tide; (d) during flood tide intrusion. After Slinger et al (1995).

in the estuary, neither its temperature nor DO were affected by the flooding. Only in the upper estuary, where the freshwater inflow had sufficient inertia to expel, low oxygen saline water did effective flushing take place.

The importance of these findings, and those subsequent to the establishment of flood tides, was that manipulation of this magnitude tends to convert the estuary into a strongly stratified system (vide p 5), and that only once a flood tidal flow is established, are the deeper older saline pockets lifted by entrainment into the fresh sea water flow. If these manipulations are timed to coincide with spring tide, maximum current velocities remove sediment from the mouth and the resultant tidal intrusion becomes the principal mechanism for flushing of the water column and the maintenance of high chemical quality in the estuary. Furthermore, where mouths close, the addition of man-made wastes will cause an already subcritical condition to become critical until either a natural flood or human intervention breaches the bar and tidal flow is re-established, but the latter is not always successful, for example in the nearby Hartenbos Estuary (Allanson et al., 1997). This investigation emphasises the need to properly understand the hydrodynamics of the estuarine system which is likely to be affected by water resource development in the river catchment.

Another particularly critical example of the effect of dam construction near to the tidal limit is found in the Kromme River at Cape St Francis (34° 09'S : 24° 52' E). The Mpofu Dam was completed in 1984 and at completion reduced an MAR of 117 $\times 10^{6}$ m^3 which had been determined for the period 1924 to 1980 to 1.3 $\times 10^{6}$. The reduced freshwater flow resulted in near homogenous axial salinity in the estuary, 35 ppt at the mouth and 30 ppt at the head, with hypersaline conditions occurring occasionally in the upper reaches in the summer months (Baird, 1999). The impacts on estuarine water quality were an increase in water transparency as freshwater inflows were normally turbid, and a decrease in 'new' nutrients via river inflow upon which phytoplankton production depends (Allanson and Read, 1995; Grange and Allanson, 1995; Scharler and Baird, 2000).

Recently, a series of important papers (Scharler and Baird, 2000; Snow et al., 2000; Strydom and Whitfield, 2000; Bate and Adams, 2000) have reported on the ecological impact of a single pulse of freshwater from the Mpofu reservoir on the freshwater starved estuary of the Kromme River. Firstly, the resulting freshwater flow of 15 $m^3 \cdot s^{-1}$ for 1.5 d flowed over the denser sea water held in the estuary, so that it had little effect on the main body of the estuary. Secondly, the pulse of freshwater rich in nitrate-N raised nitrate levels in the partially mixed shallower seaward section of the estuary; but these were short-lived. On the other hand, SRP concentrations were lowered because the freshwater pulse failed to introduce an adequate amount of new phosphate. Consequenly, the Redfield ratio rose to >320, indicating a striking phosphorus limitation in the estuary. Thirdly, it demonstrated unequivocally that freshets of this magnitude delivered from the reservoir, and in this case, equalling the volume of water required per annum to compensate for evaporation (Jezewski and Roberts, 1986), did not replace the biological impact the base flow would have provided if the river flowed naturally into its estuary. Confirmation of this feature of estuarine management has been given by Wooldridge and Scharler (1999) who have shown that the adjacent Gamtoos Estuary, receiving slightly less than 1×10^6 $m^3 \cdot a^{-1}$ as a sustained base flow, possessed all the facies of a healthy estuary in the level of phytoplankton production, and a rich zooplankton and benthic macrofauna.

Bate and Adams (2000), while recording these features of the Gamtoos Estuary, drew attention to the differences in the activities in their respective catchments. Thus, while the Gamtoos Estuary receives water from heavily cultivated lands, the water which flows into the Mpofu reservoir derives from a catchment in which there is little agriculture other than grazing. The difference in mineral nutrients, which has been noted by this comparison, does point to the danger of simple extrapolation.

Nevertheless, recognising the need for a base flow in the Kromme Estuary, Bate and Adams (2000) have estimated the flow necessary to reactivate its chemical and biological compartments. They estimate a base flow of 0.77 $m^3 \cdot s^{-1}$ in order to allow optimal phytoplankton productivity. This is very much greater than the base flow provided by the existing evaporation allocation of 2×10^6 $m^3 \cdot a^{-1}$ or 0.06 $m^3 \cdot s^{-1}$ which represents 8% of the optimum!

Interbasin transport

Interbasin transport is one of the most effective means of water transfer to water-poor areas or where demand exceeds the supply of the respective catchment, but the impact of this engineered hydrology upon the estuaries of either donor or recipient rivers is poorly understood. The estuary of the Great Fish River, which receives water from the distant Orange River via a number of irrigation schemes, is presently characterised by a sustained river flow of 5 to 15 $\times 10^6$ m per month. Prior to this change the estuary was largely marine-dominated with occasional episodic floods which cleared out the accumulation of silt and sands. Allanson and Read (1995) have reported on the high nutrient loads, 369 $t \cdot d^{-1}$ of nitrate-N and 219 $t \cdot d^{1}$ of phosphate-P which the estuary now receives at the onset of the summer floods. The result is that N:P ratios of 21:1 are common, and even though the incoming water is turbid, the turbulence set up in the shallow water column allows the maintenance of high concentrations (20.5 $\mu g \cdot \ell^{-1}$) of algal chlorophyll (Grange and Allanson, 1995).

Biogeochemical processes

Without exception, estuaries exhibit biogeochemical activity. Most of the inorganic substances fixed by plants and converted to organic form are deposited in their sediments. Furthermore, the effectiveness of these sediments in organic matter degradation and nutrient cycling is inversely related to the depth of the water column (Nedwell et al., 1999). This implies that in shallow microtidal estuaries with extensive intertidal mudflats and wetlands, the submerged sediments and intertidal wetlands are the principal areas of POM deposition, benthic primary production, breakdown of organic matter and nutrient cycling, (Mitsch and Gosselink 1993; Allanson and Winter 1999).

These complex processes depend upon the sediments acting as sinks for oxygen, sulphate and nitrate which are used as electron acceptors by specific groups of bacteria in the bottom sediments during the processing of nitrogenous and phosphorus compounds, (Nedwell et al., 1999; Beck and Bruland, 2000). Bacterially-mediated processes generate nutrients largely through mineralisation, and the flux of the nitrogenous nutrients, e.g. nitrate and ammonium from the wetland sediments depend upon the vertical concentration gradient of each solute across the sediment-water interface (Taylor 1992; Nedwell et al.,1999) and the temporal and spatial integrity of the oxic layer at this interface (Fig. 5a). Nedwell et al. (1999) report that this layer may vary in vertical extent seasonally and along the estuary depending upon the organic matter available.

The variability of the oxic layer and, therefore, the accessibility of the deeper anaerobic sediment layers influence the availability

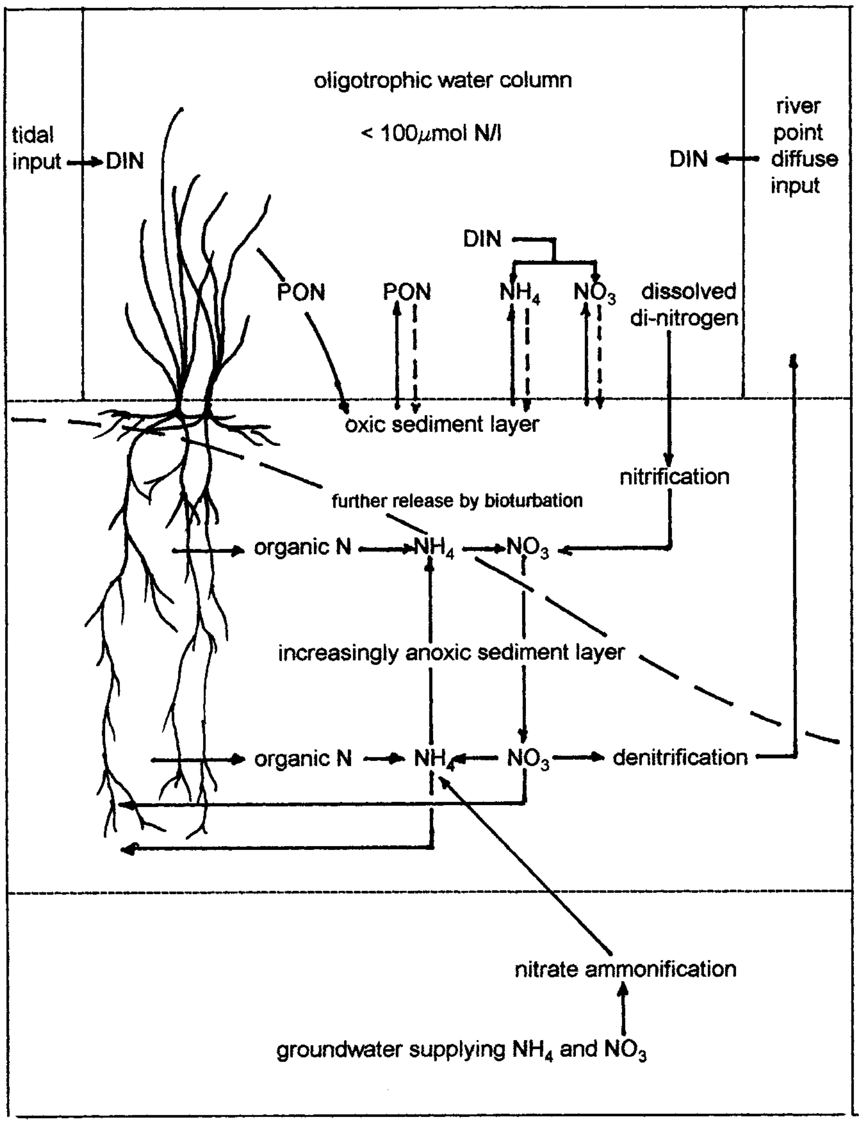

Figure 5a
A summary of processes responsible for nutrient exchange in eelgrass meadows in an oligotrophic estuary: (a) nitrogen and (b) phosphorus. The curved dashed line within the sediment compartment represents the variable transition depth between oxic and anoxic sediments within the meadows, and along the estuary. The relative sizes of the compartments have no quantitative meaning. Partly constructed from Valiela & Teal (1979), Mitsch & Gosselink (1993).

of phosphorus. This element is one of the limiting nutrients to primary production both in the water column and on the sediment surface, although there would appear to be some controversy when applied to estuaries worldwide (Nixon, 1980). Nevertheless, it is well-known that redox potential is one of the most important factors influencing phosphorus exchange. Changes in redox potential and pH alter the quantity of phosphorus held in association with charged cations, primarily Fe, Al, clay and floc particles, (Froelich, 1988). Thus, deoxygenation results in the release of phosphorus from sesquioxide complexes at Eh values below -200 mV. At lower potentials, H_2S reacts with the complexes to liberate further phosphorus.

Phosphorus release rates of 2.5 mg P $m^{-2} \cdot d^{-1}$ were reported by Silberbauer (1982) when sediment cores taken from Swartvlei were held anaerobic, and lower rates of 1.6 mg P $m^{-2} \cdot d^{-1}$ were obtained under aerobic conditions. Development of this investigation by Howard-Williams and Allanson (1981) in the *Potamogeton pectinatus* canopy in Swartvlei estuary using ^{32}P demonstrated that the soluble fractions of phosphorus in the water column are taken up rapidly in the canopy, and that the sediments, including the detrital layer are relatively much slower, although they possess large nutrient reserves. The ability of the littoral zone to sequester essential nutrients points to the vital role it plays in nutrient pathways.

A similar flux for phosphorus, but involving excretion by the eelgrass *Zostera capensis* following the earlier findings of McRoy et al. (1972) for *Z. marina* in North America, but now involving three obvious metabolic compartments, has been defined for the Swartvlei Estuary by Liptrot (1978). In this context phosphorus is cycled between *Zostera* beds and floating mats of the alga *Enteromorpha* spp. (Fig. 5b). During the lagoon phase, the overall effect is to retain the original phosphorus in the estuary by depleting the accumulated phosphorus in the sediments through transfer to and excretion from *Zostera* to the *Enteromorpha* mats, and in such a way that the water column retains a phosphorus equilibrium of 22 to 26 $\mu g \cdot \ell^{-1}$. Similar equilibrium values have been reported by Pomeroy et al. (1965) in the Doboy Sound and in the Tamar Estuary by Butler and Tibbetts (1972).

Trace metals

The trace element status of South Africa's estuaries has been surveyed in considerable detail by Watling and Watling (1983). Their general conclusion is that the southern and eastern Cape estuaries could not be considered to be polluted. And while the effect of trace metals on metabolic processes is appreciated, little

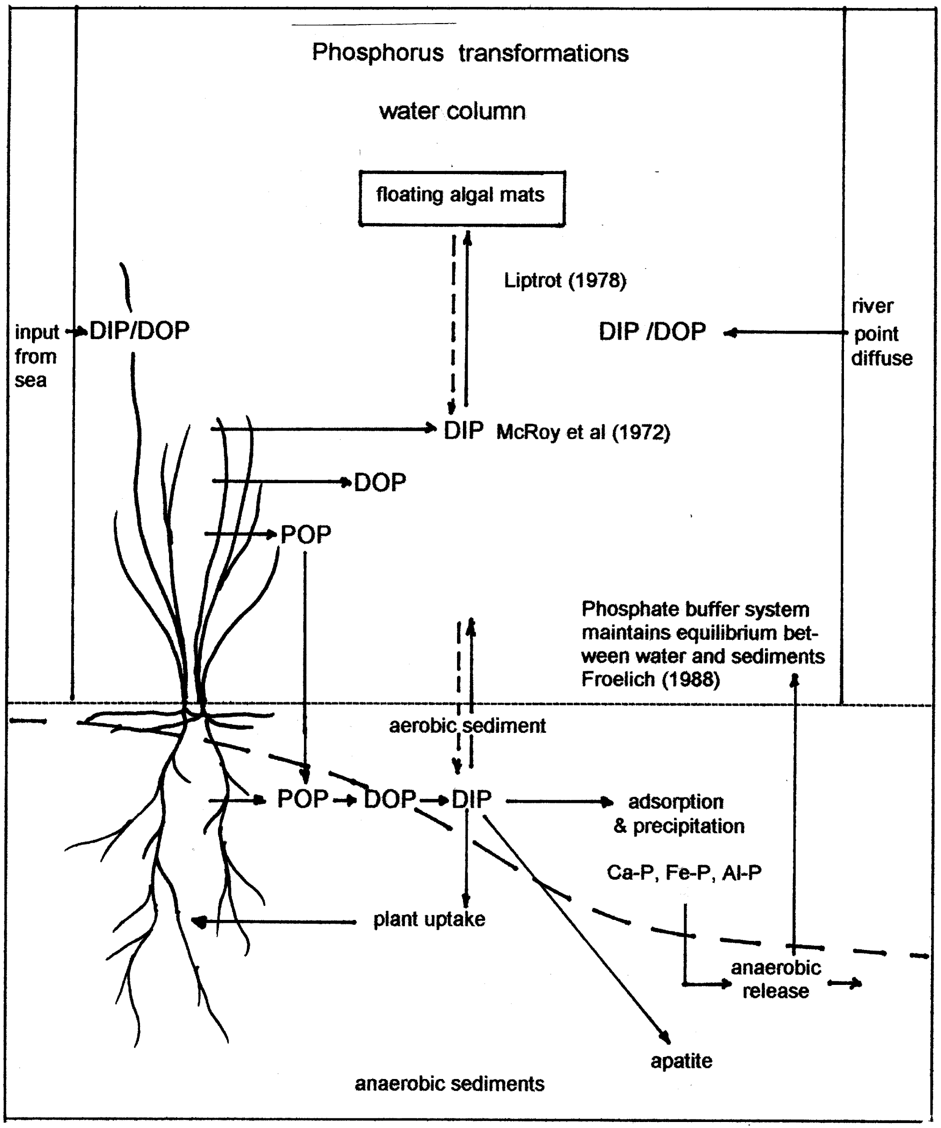

Figure 5 b

direct investigation has been carried out in South Africa with the exception of Talbot's (1988) studies on the influence of zinc, copper and lead on nitrogen metabolism in the coastal marine environment: zinc, it appears, has the greatest impact on ammonification and nitrification, followed by copper and lead. Howarth et al. (1988) in a review of biogeochemical controls of nitrogen fixation in freshwater and marine systems have shown, that while the concentration of molybdenum in oxic sea water is higher than in freshwater, its availability is lower since sulphate inhibits molybdate assimilation by planktonic algae and bacteria and, therefore, nitrogen fixation.

Influence of macrofauna

The water quality of estuaries is further modified by the macrofauna. Predation, filter and particle-feeding and scavenging as well as microbial assimilation, cause new organic compounds and CO_2 to be released into the water. The intertidal sediments and their communities are of particular importance in this regard. Recent work by Taylor (1992) has determined the role of the littoral zone in modifying concentrations of 'new' nutrients derived from the sea, particularly during periods of upwelling. These data show that both tidal creek and salt marsh, when inundated with upwelled water, had a marked effect on the net flux of nitrate, but no change of any significance could be detected for other nutrients. In the particular experiments, the mean net uptake was 7 196 sd 101 $\mu g\ N{\cdot}m^{-2}$ per tide (n = 6), for the tidal creek and 3 164 sd 164 $\mu g\ N{\cdot}m^{-2}$ per tide. The marked difference between the two sites is held by Taylor (1992) to be due to the moister, more reduced sediments (Eh <200 mV) and the likelihood of denitrifying assemblages in the tidal creek.

Taylor and Allanson (1993; 1995) have determined quantitatively not only the fluxes of DOC, POC and TOC between the marsh and the estuary, but also the very significant modulation imposed by the benthic marsh crabs, *Sesarma catenata* and *Paratylodiplax edwardsii*. The conclusion from these data is that the two crab species enhanced the net loss of carbon from the marsh, although they did so in quite different ways. *P.edwardsii* enhances the net loss by increasing TOC flux, while *S. catenata* does so by reducing epibenthic primary production.

Further faunal effects are even more surprising. Gerdol and Hughes (1994) found that the removal of the amphipod, *Corophium volutator* from surface sediments led to a more stable substratum due to, they concluded, a build-up of leucopolysaccharides secreted by the now more abundant microflora, and which acts as a binding agent in the sediment. The absence of the amphipod also decreased the level of denitrification, while its presence caused a 3-fold increase in denitrification within the sediment and a 5-fold increase in denitrification of nitrate derived from the overlying water.

Likewise, Harding (1994) implicated the polychaet worm *Ficopomatus enigmaticus* in Sandvlei near Cape Town, of contributing towards the maintenance of water quality by reducing suspended particle concentration at a rate of *ca*.130 kg (wet mass) per hour. This activity contributed to the success of a number of recreational activities in the vlei in spite of an earlier near collapse of its ecological structure as a result of polluted discharges in the catchment of the vlei.

The role of soft bottom tube dwellers in estuaries has received some attention. Branch & Pringle (1987) determined that the activities of the tube-dwelling prawn *Callianassa krausii* results in the bioturbation of about 12 kg sediment $m^{-2} \cdot d^{-1}$. The effect this has on increasing the redox potential to >200 mV is important in reducing the expansion of anaerobic sediments in estuaries. Law et al. (1991) reported a strong covariance between rates of denitrification and the degree of bioturbation by the polychaet worm *Nereis diversicolor* and ascribed this to increased transport and supply of nitrate via the burrows. In the Tamar Estuary, England, Davey and Watson (1995) found that irrigation by *N.diversicolor* of its burrow accounted for fluxes of ammonium and nitrate to the water column of between 10 and 20 times the measured input derived from riverine and sewage sources.

In a South African study Tibbles et al. (1994) found that oxygen stimulation of nitrogenase activity was less marked in the sediment lining of *Upogebia africana* and *Callianassa kraussi* burrows, notwithstanding the increased aeration of the burrow, if the microaerophilic or aerobic diazotrophs are displaced from around the burrow where they normally occur.

It is accepted that the intertidal sediments and their biota, whether on the surface, burrowing or living within the sediment interstices, are important components involved in the external metabolism of organic and inorganic materials introduced either via the river, anthropogenic activities or the sea. (Valiela and Teal 1979; Nixon 1980; Fisher et al., 1982; Kokkinn and Allanson 1985; Tibbles et al.; 1994 and Herman et al., 1999).

Thus, where the geomorphological evolution of an estuary has provided extensive intertidal mudflats, eelgrass meadows and saltmarshes, and these have not been encroached upon through human activities, the zone between the tides becomes the focus of significant metabolic activities divided between two large compartments. Firstly, the anaerobic system with its propensity to convert organic materials to carbon, nitrogen and sulphur hydrides, CO_2, N_2O and N_2, and secondly, a sediment /water interface which partially seals the anaerobic engine beneath, while providing the essential aerobic atmosphere and sediment structure for successful colonisation of a host of post-larvae and germinating seeds. The porous nature of the intertidal sediments makes for a leaky sieve allowing exit and entrance (ably assisted by bioturbation) to the diverse chemical materials generated within the marsh and received by it from the inundating water flow. This is illustrated diagrammatically in Fig. 5a,b for nitrogen and phosphorus.

Applying these biogeochemical principles in estuaries subject to seasonal closure, the sequestration of phosphorus and recycling of nitrogen will maintain the metabolic processes, and prevent irreversible chemical and, therefore, biological change. The addition, however, of high loadings of allochthonous nutrients in the form of treatment works effluent as occurs in the Hartenbos Estuary (34° 07' S:22° 07' E) which only opens to the sea infrequently, overwhelms these natural processes and hyper-eutrophication results (Allanson et al., 1997). A particular distressing consequence is the accumulation of ammonia and free sulphides in the water column causing severe deterioration in water quality, and decreasing biological diversity.

Some modelling considerations

With such a diversity of estuarine types and individual response to physical and chemical determinants in the highly dynamic environment of South African estuaries, the need for techniques with which to integrate these features, into conceptual and numerical models, is becoming essential. Taljaard and Slinger (1997) have found that numerical modelling is used most appropriately in compliance testing. This allows the fate of different water quality constituents to be determined and the ambient variability of water quality parameters to be assessed. For example, it has been useful in determining by simple regression, the association of dissolved nutrient distributions to salinity in both the Berg and Swartkops Estuaries.

Slinger et al. (1998) have, through the support of the Water Research Commission, examined the capability of a one-dimensional water quality module, Mike 11, developed by the Danish Hydraulic Institute. They concluded that the one-dimensional module performed well in modelling the salinity distribution, thermal variations and dissolved oxygen concentrations both in the Great Berg and Swartkops Estuaries, and seems to be well-suited to application on permanently open South African estuaries.

Dissolved nutrient levels have not been modelled using the Mike 11 system, for two reasons: the linear association of nutrients distributions to salinity under certain flow conditions, e.g. winter flows in the Great Berg Estuary, could be predicted more accurately with simple regression techniques than a full modelling application, and no conceptual model of the biogeochemical processes operative in estuaries could be derived from available data. This is, in itself, surprising, as there is vast international literature providing quantitative data and analysis of extensive investigations on biogeochemical cycles affecting not only nitrogen and phosphorus, but carbon, silica and trace metals throughout an array of salt marshes and inter-tidal creeks. (Valiela and Teal 1979; Nixon 1980; Howarth et al.,1988; Nowicki and Oviatt 1990; Taylor 1992; Tibbles et al., 1994). Without exception, they have demonstrated that inputs of DON and PON intercepted by wetlands undergo various biological transformations which results in the initial loading being exported from the marsh as ammonium, PON, derived from plant decay and di-nitrogen. Both Valiela & Teal (1979) and Nowicki and Oviatt (1990) stress the balance which exists in either a mature marsh or in experimental mesocosms between imports and exports.

The difficulty we have in integrating conceptual models of biogeochemical processes into a more holistic system, is that all too often the advective processes, either from the sea or from the river and other catchments drainage, e.g. storm water flows, dominate the influence of biogeochemical processes. Thus, the rate of adjustment of the N:P concentration ratio (Howarth et al., 1988) will be slow relative to, for example, the advective throughput of nutrients. We have come to expect that estuaries have shorter residence times than, for example, lakes and a consequence is that the estuarine water tends to be nitrogen limited because the nitrogen fixed is rapidly swept away. In *oligotrophic* systems, the inter-tidal marshes and creeks are not rich sources of nitrogen supply to the water column. Equally important is the impact of reduction in river flow to the estuary. The persistent reduction in nitrogen, phosphorus, silica and POM which dams bring about, significantly alter the transfer of energy within the affected system. Baird and Heymans (1996) have clearly demonstrated this impact in the Kromme Estuary,

What is required in the microtidal estuaries of South Africa is a concerted effort to establish nitrogen and phosphorus budgets so that models of nutrient dynamics, and the all important impact of anthropogenic inputs may be quantified, using as a starting point conceptual models like those illustrated in Fig. 5a, b in which the principal pathways through which nitrogen and phosphorus flow in an estuary are given.

In many South African estuaries, however, human contact in its diverse forms is often difficult to distinguish from ambient variability in the short term (Slinger et al., 1998). The need for long-term sequences of environmental data is, therefore, paramount if human impacts are to be separated from natural variability. Furthermore, it is becoming more readily recognised that the water column does not always exhibit startling change: human impacts are more sensitively reflected in the diversity and productivity of the plants and animals which inhabit the intertidal and subtidal habitats, and these changes may be more significant in the long term, (Allanson et al., 2000). Examples of this approach are given by Slinger et al. (1998) and Morant and Quinn (1999) who have developed models which draw extensively upon the rich diversity of new biological and ecological knowledge about the communities of estuaries, and on the impact, for example, of mouth closure upon estuarine biota, particularly those with planktonic migratory larvae.

Global warming and coastal marsh survival

Estuaries with wide gently shelving inter-tidal shores, e.g. Great Brak River, Wilderness and Knysna are examples which Hughes and Brundrit (1992) consider, after risk analysis, to be the first in line to feel the impact of rising sea level as a consequence of global warming. Crooks and Turner (1999) recognise that if coastal marshes are to maintain elevation sufficient for saltmarsh survival, and to avoid "ecological drowning", accretion of the surface will have to keep pace with sea level rise. As more rivers are regulated, the supply of sediments (silts and clays) becomes less so that such an equilibrium becomes unsustainable and the coastal marshes disappear.

Nutrient ratios

Of equal importance, and relating directly to the exponential increase in river regulation worldwide, is the serious disturbance of the ratios of nutrient elements silica: nitrogen: phosphorus in river inflows to estuarine and coastal ecosystems.(Justic et al., 1995; Conley, 2000; Ittekkot et al., 2000). There is increasing evidence that the silicate supply to these coastal water bodies is derived from river flow, and that river regulation by dams or the inter-basin transfer of water is resulting, particularly in enclosed seas e.g. the Black Sea, in a reduction of 80% of observed silicate (Humborg et al.,1997) and with it a change in phytoplankton species array involving a decrease in diatoms and an increase in bloom-forming non-siliceous algal species which are often toxic. Allanson et al., (1999) drew attention to the possibility that the frequent and intense toxic phytoplankton blooms in South African coastal waters may be related to a reduction in silicate as a consequence of the substantial flow regulation which has influenced the Orange, Vaal, Berg and Olifants Rivers during the past 50 years. If nothing else, they emphasised that this possibility represents an important research challenge!

Acknowledgements

The valuable and critical review by two independent reviewers is warmly acknowledged. To the Master and Fellows of St Catharine's College, Cambridge, where the revision of the MS was done, and the Ernest Oppenheimer Memorial Trust and Rhodes University for financial support while in Cambridge, my grateful thanks.

References

ALEXANDER WJR (1985) Hydrology of low latitude Southern Hemisphere landmasses. In: Davies BR and Walmsley RD (eds.) *Perspectives in Southern Hemisphere Limnology*. Kluwer Academic Publishers, Dordrecht. 75-83.

ALLANSON BR and WINTER PED (1999) Chemistry. In: Allanson BR and Baird D (eds.) *Estuaries of South Africa.* Cambridge Univ. Press. 53-90.

ALLANSON BR, BAIRD D and HEYDORN A (1999) Perspectives. In: Allanson BR and Baird D (eds.) *Estuaries of South Africa.* Cambridge Univ. Press. 321-328.

ALLANSON BR and HOWARD-WILLIAMS C (1984) A contribution to the chemical limnology of Swartvlei. *Arch. Hydrobiol.* **99** 133-159.

ALLANSON BR, GRANGE N and MAREE B (1997) Specialist investigation: Environmental Implications for the Hartenbos Estuary of Upgrading the Existing Sewage Treatment Works (A report prepared for Alan Wijnberg Inc, Consulting Port, Coastal and Environmental Engineers, Mossel Bay.) Allanson Grange Associates, Consulting Aquatic Ecologists and Environmental Scientists, Knysna. 28 pp.

ALLANSON BR, NETTLETON J and DE VILLIERS C (2000) Intertidal benthic macrofauna richness and diversity in the Knysna estuary: A 50 year comparison. *Trans. Roy. Soc. S. Afr.* **55** 141-162.

ALLANSON BR and READ GHL (1995) Further comment on the response of Eastern Cape Province estuaries to variable freshwater inflows. *S. Afr. J. Aquat. Sci.* **21** 56-70.

ALLANSON BR and VAN WYK JD (1969) An introduction to the physics and chemistry of some lakes in northern Zululand. *Trans. Roy. Soc. S. Afr.* **38** 217-240.

BAIRD D (1999) Estuaries as ecosystems: A functional and comparative analysis. In: Allanson BR and Baird D (eds.) *Estuaries of South Africa.* Cambridge Univ. Press. 269-288.

BAIRD D and HEYMANS JJ (1996) Assessment of ecosystem changes in response to freshwater inflow in the Kromme River estuary, St Francis Bay, South Africa: A network analysis approach. *Water SA* **22** 1-10.

BATE GC and ADAMS JB (2000) The effects of a single freshwater release into the Kromme Estuary. 5: Overview and interpretation for the future. *Water SA* **26** 329-332.

BECK NG and BRULAND KW (2000) Dill biogeochemical cycling in a hyperventilating shallow estuarine environment. *Estuary* **23** 177-187.

BRANCH GM and DAY JA (1984) Ecology of southern African estuaries: Part 13. The Palmiet River estuary in the south-western Cape. *S. Afr. J. Zoo.* **19** 63-77.

BRANCH GM and PRINGLE A (1987) The impact of the sand prawn *Callianassa kraussi* Stebbing on sediment turnover and on bacteria, meiofauna, and benthic microfauna. *J. Exp. Mar. Biol.* **107** 219-235.

BUTLER EJ and TIBBETTS S (1972) Chemical survey of the Tamar estuary. I. Properties of the waters. *J. Mar. Biol. Assoc. U.K.* **52** 681-699.

CONLEY DJ (2000) Biogeochemical nutrient cycles and nutrient management strategies. *Hydrobiol.* **410** 81-96.

COOPER A, WRIGHT I and MASON T (1999) Geomorphology and sedimentology. In: Allanson BR and Baird D (eds.) *Estuaries of South Africa.* Cambridge Univ. Press. 5-26.

CROOKS S and TURNER RK (1999) Integrated coastal management: Sustaining estuarine natural resources. In: *Advances in Ecological Research, Estuaries* **29** 241-289. Academic Press, London.

DAVEY JT and WATSON PG (1995) The activity of *Nereis diversicolor* (Polychaeta) and its impact on nutrient fluxes in estuarine waters. *Ophelia* **41** 57-70.

DAY JH (ed.) (1981) *Estuarine Ecology with Particular Reference to Southern Africa*. Balkema, Cape Town. 411 pp.

FISHER TR, CARLSON PR and BARBER RT (1982) Sediment nutrient regeneration in three North Carolina estuaries. *Estuarine Coastal Shelf Sci.* **14** 101-106.

FROELICH PN (1988) Kinetic control of dissolved phosphate in natural rivers and estuaries: A primer on the phosphate buffer mechanism. *Limnol. Oceanogr.* **33** 649-68.

GERDOL V and HUGHES RG (1994) Effect of *Corophium volutator* on the abundance of benthic diatoms, bacteria and sediment stability in two estuaries in southeastern England. *Mar. Ecol. Prog. Ser.* **114** 109-15.

GRANGE N and ALLANSON BR (1995) The influence of freshwater inflow on the nature, amount and distribution of seston in estuaries of the Eastern Cape, South Africa. *Estuarine Coastal Shelf Sci.* **40** 402-420.

GRANGE N, WHITFIELD AK, DE VILLIERS CJ and ALLANSON BR (2000) The response of two South African east coast estuaries to altered river flow regimes. *Aquatic Conserv. Mar. Freshwater Ecosyst.* **10** 155-177.

HARDING WR (1994) Water quality trends and the influence of salinity in a highly regulated estuary near Cape Town, South Africa. *S. Afr. J. Sci.* **90** 240-246.

HERMAN PMS, MIDDELBURG JJ, VAN DE KOPPEL J and HEIP CHR (1999) Ecology of estuarine macrobenthos. In: Nedwell DB and Raffaelini DG (eds.) *Advances in Ecological Research: Estuaries* **29** 195-231. Academic Press, London.

HOBDAY DK (1979) Geological evolution and geomorphology of the Zululand coastal plain. In: BR Allanson (ed.) *Lake Sibaya*. Monographiae Biologicae 36. Dr W Junk publishers, The Hague. 1-16.

HOWARD-WILLIAMS C and ALLANSON BR (1981) Phosphorous cycling in a dense *Potamogeton pectinatus* L. bed. *Oecol.* **49** 56-66.

HOWARTH RW, MARINO R and COLE JJ (1988) Nitrogen fixation in freshwater, estuarine and marine ecosystems. 2. Biogeochemical controls. *Limnol. Oceanogr.* **33** 688-701.

HUGHES P and BRUNDRIT GB (1992) An index to assess South Africa's vulnerability to sea level rise. *S. Afr. J. Sci.* **88** 308-311.

HUMBORG C, ITTEKOT V, COCASU A and VON BODUNGEN B (1997) Effect of Danube River Dam on Black Sea biogeochemistry and ecosystem structure. *Nature* **386** 385-388.

ITTEKKOT V, HUMBORG C and SCHÄFER P (2000) Hydrological alterations and marine biogeochemistry: A silicate issue. *Bioscience* **50** 776-782.

JEZEWSKI WA AND ROBERTS CPR (1986) Estuarine and Lake Freshwater Requirements. Department of Water Affairs Technical Report No TR129.

JUSTIC D, RABELAIS NN, TURNER RE and DORTCH Q (1995) Changes in nutrient structure of river-dominated coastal waters: Stoichiometric nutrient balance and its convergence. *Estuarine Coastal Shelf Sci.* **40** 339-56.

KING LC (1972) *The Natal Monocline: Explaining the Origin and Scenery of Natal, South Africa.* Univ. of Natal, Durban.

KOKKINN MJ and ALLANSON BR (1985) On the flux of organic carbon in a tidal salt marsh, Kowie River estuary, Port Alfred, South Africa. *S. Afr. J. Sci.* **81** 613-17.

LARGIER JL and SLINGER JH (1991) Circulation in highly stratified Southern African estuaries. *S. Afr. J. Aquat. Sci.* **17** 103-115.

LARGIER JL and TALJAARD S (1991) The dynamics of tidal intrusion, retention, and removal of seawater in a bar-built estuary. *Estuarine Coastal Shelf Sci.* **33** 325-338.

LAW CS, REES AP and OWENS NJP (1991) Temporal variability of denitrification in estuarine sediments. *Estuary Coast Shelf Sci.* **33** 37-56.

LIPTROT MRN (1978) Community Metabolism and Phosphorus Dynamics in a Seasonally Closed South African Estuary. M.Sc. Thesis. Rhodes Univ., Grahamstown.

MACKAY HM (1993) The Impact of Urban Runoff on the Water Quality of the Swartkops Estuary: Implications for Water Quality Management. WRC Report KV 45/93.

McROY CP, BARSDALE RJ and NEBERT M (1972) Phosphorus cycling in an eelgrass (*Zostera marina* L.) ecosystem. *Limnol. Oceanogr.* **17** 58-67.

MITSCH WJ and GOSSELINK JG (1993) *Wetlands* (2nd edn.) i-xii , Van Nostrand Reinhold, New York. 722 pp.

MORANT PD and QUINN NW (1999) Influence of Man and management of South African estuaries. In: Allanson BR and Baird D (eds.) *Estuaries of South Africa.* Cambridge Univ. Press. 289-320.

NEDWELL DB, JICKELLS JD, TRIMMER M and SANDERS R (1999) Nutrients in estuaries. In: *Advances in Ecological Research: Estuaries* **29** 142-192.

NIXON SW (1980) Between coastal marshes and coastal wetlands - A review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. In: Hamilton P and MacDonald K (eds.) *Estuaries and Coastal Processes with Emphasis on Modelling.* Plenum Publishing Corporation, New York.

NOWICKI BL and OVIATT CA (1990) Are estuaries traps for anthropogenic nutrients - Evidence from estuarine mesocosms. *Mar. Ecol. Prog. Ser.* **66** 131-146.

ORME AR (1973) Barrier and Lagoonal Systems along the Zululand Coast, South Africa. Office of Naval Research, Technical Report 1.

PETHICK JS (1984) *An Introduction to Coastal Geomorphology*. Edward Arnold, London. 260 pp.

PETHICK JS (1992) Saltmarsh geomorphology. In: Allen JRL and Pye K (eds.) *Saltmarsh Morphodynamics, Conservation and Engineering Significance*. Cambridge Univ. Press, Cambridge. 41-62.

POMEROY LR, SMITH EE and GRANT CM (1965) The exchange of phosphate between estuarine water and sediments. *Limnol. Oceanogr.* **10** 167-172.

RAMM AEL (1992) Aspects of the biogeochemistry of sulphur in Lake Mpungwini, Southern Africa. *Estuarine Coastal Shelf Sci.* **34** 253-61.

REDDERING JSV (1988) Prediction of the effects of reduced river discharge on the estuaries of the south-eastern Cape Province, South Africa. *S. Afr. J. Sci.* **84** 726-730.

ROBARTS RD and ALLANSON BR (1977) Meromixis in the lake-like upper reaches of a South African estuary. *Arch. Hydrobiol.* **80** 531-40.

SCHARLER UM and BAIRD D (2000) The effects of a single freshwater release into the Kromme Estuary: 1: General description of the study area and physico-chemical responses. *Water SA* **26** 291-300.

SCHARLER UM, BAIRD D and WINTER PED (1998) Diversity and Productivity of Biotic Communities in Relation to Freshwater Inputs in Three Eastern Cape Estuaries.WRC Report No 463/1/98.

SCOTT KMF, HARRISON AD and MACNAE W (1952) The ecology of South African estuaries. Part 2: the Klein River estuary, Hermanus, Cape. *Trans. Roy. Soc. S. Afr.* **33** 282-331.

SILBERBAUER MJ (1982) Phosphorus dynamics in the monimolimnion of Swartvlei. *J. Limnol. Soc. S. Afr.* **8** 54-60.

SLINGER JH and TALJAARD S (1994) Preliminary investigation of seasonality in the Great Brak estuary. *Water SA* **20** 279-288.

SLINGER JH, TALJAARD S and LARGIER J (1995) Changes in estuarine water quality in response to a freshwater flow event. In: Dyer KR and Orth RJ (eds.) *Changes in Fluxes in Estuaries*. Olsen & Olsen, Denmark. 51-56.

SLINGER JH, TALJAARD S, ROSSOUW M and HUIZINGA P (1998) Water Quality Modelling of Estuaries. W RC Report No 664/1/98.

SNOW GC, BATE GC and ADAMS JB (2000) The effects of a single freshwater release into the Kromme Estuary 2: Microalgal response. *Water SA* **26** 301-310.

STRYDOM NA and WHITFIELD AK (2000) The effects of a single freshwater release into the Kromme Estuary 4: Larval fish response. *Water SA* **26** 319-328.

TALBOT MMJ (1988) Trace Metals in the Marine Coastal Environment and their Effect on Nitrogen Recycling. Ph.D. thesis. Univ. of Port Elizabeth, Port Elizabeth 258pp.

TALJAARD S (1987) Nutrient Circulation in the Palmiet River Estuary: A Summer Study. M.Sc. Thesis,Univ. of Port Elizabeth, Port Elizabeth.

TALJAARD S and LARGIER JL (1989) Water Circulation and Nutrient Distribution Patterns in the Palmiet River Estuary: A Winter Study. CSIR Research Report No 680. Stellenbosch, CSIR.

TALJAARD S and SLINGER JH (1997) Decision Support Requirements for Estuarine Water Quality Management. CSIR Technical Report ENV/S-T 97001, Stellenbosch.

TALJAARD S, DE VILLIERS S, FRICKE AH and KLOPPERS WS (1992) Water Quality Status of the Rietvlei/Milnerton Lagoon System (Diep River Estuary). Data Report EMAS-D 92007. CSIR, Stellenbosch

TAYLOR DI (1992) The influence of upwelling and short-term changes in concentrations of nutrients in the water column on fluxes across the surface of a salt marsh. *Estuary* **15** 68-74.

TAYLOR DI and ALLANSON BR (1993) Impacts of dense crab populations on carbon exchanges across the surface of a salt marsh. *Mar. Ecol. Prog. Ser.* **101** 119-129.

TAYLOR DI and ALLANSON BR (1995) Organic carbon fluxes between a high marsh and estuary, and the applicability of the Outwelling Hypothesis. *Mar. Ecol. Prog. Ser.* **126** 263-270.

TIBBLES BJ, LUCAS MI, COYNE VE and NEWTON ST (1994) Nitrogenase activity in marine sediments from a temperate salt marsh lagoon. Modulation by complex polysaccharides, ammonium and oxygen. *J. Exp. Mar. Biol.* **184** 1-20.

VALIELA I and TEAL JM (1979) The nitrogen budget of a saltmarsh ecosystem. *Nature* **280** 652-656.

WATLING RJ and WATLING HR (1983) Metal surveys in South African estuaries. VII. Bushmans, Kariega, Kowie and Great Fish Rivers. *Water SA* **9** 66-70.

WHITFIELD AK and BRUTON MN (1989) Some biological implications of reduced freshwater inflow into Eastern Cape estuaries: A preliminary assessment. *S. Afr. J. Sci.* **85** 691-694.

WOOLDRIDGE TH (1976) The zooplankton of the Msikaba estuary. *Zool. Afr.* **11** 23-44.

WOOLDRIDGE TH and CALLAHAN R (2000) The effects of a single freshwater release into the Kromme Estuary. 3: Estuarine zooplankton response. *Water SA* **26** 311-318.